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Reefcrete: reducing the environmental footprint of concretes for eco-engineering marine structures

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Abstract

The ecological value of engineered marine structures can be enhanced by building-in additional habitat complexity. Pre-fabricated habitat units can be cheaply and easily cast from concrete into heterogeneous three-dimensional shapes and surface topographies, with proven ability to enhance biodiversity on artificial structures. The net ecological benefits of enhancement using concrete, however, may be compromised on account of its large environmental footprint and poor performance as substrate for many marine organisms. We carried out a pilot study to trial alternative cast-able "*Reefcrete*" concrete mixes, with reduced environmental footprints, for use in the marine environment. We used partial replacement of Portland cement with recycled ground granulated blast-furnace slag (GGBS), and partial replacement of coarse aggregate with hemp fibres and recycled shell material. We calculated the estimated carbon footprint of each concrete blend and deployed replicate tiles in the intertidal environment for 12 months to assess their performance as substrate for marine biodiversity. The hemp and shell concrete blends had reduced carbon footprints compared to both ordinary Portland cement based concrete and the GGBS based control concrete used in this study. At the end of the experiment, the hemp and shell blends supported significantly more live cover than the standard GGBS control blend. Taxon richness, particularly of mobile fauna, was also higher on the hemp concrete than either the shell or GGBS control. Furthermore, the overall species pool recorded on the hemp concrete was much larger. Community compositions differed significantly on the hemp tiles, compared to GGBS controls. This was largely explained by higher abundances of several taxa, including canopy-forming algae, which may have facilitated other taxa. Our findings indicate that the alternative materials trialled in this study provided substrate of equal or better habitat suitability compared to ordinary GGBS based concrete. Given the growing interest in

ecological engineering of marine infrastructure, we propose there would be great benefit in further development of these alternative "*Reefcrete*" materials for wider application.

Key words: Biodiversity enhancement; Blue-green infrastructure; Carbon footprint; Ecological engineering; Green concrete; Ocean sprawl

1. Introduction

"Ocean sprawl" is causing considerable damage to the ecological condition and functioning of marine and coastal habitats globally (see recent review by Firth et al., 2016b). In addition to causing habitat loss and fragmentation, engineered structures in the marine environment are known to support low biodiversity and 'non-natural' communities of marine life compared to natural rocky habitats (Aguilera et al, 2014; Chapman, 2003; Chapman and Bulleri, 2003; Firth et al., 2013; Moschella et al., 2005), often harbouring non-native and invasive species (Airoldi et al., 2015; Bulleri and Airoldi, 2005; Glasby et al., 2007; Mineur et al., 2012; Tyrrell and Byers, 2007). The field of ecological engineering has emerged to investigate ways of enhancing the ecological value of artificial structures, in an effort to maximise their potential to support biodiversity and natural capital. Researchers have approached this by trialling a variety of engineering manipulations to increase topographical complexity at varying scales, to build-in refuge and habitat niches that are often absent from engineered structures (reviewed by Firth et al., 2016b).

The addition of topographic complexity such as surface texture, cracks, holes and pools has been shown to be an effective means of promoting biodiversity on artificial marine structures (Chapman and Blockley, 2009; Evans et al., 2016; Firth et al. 2014, 2016a; Martins et al., 2010; Paalvast, 2015; Perkol-Finkel and Sella, 2016; Sella and Perkol-Finkel, 2015). Large-scale pre-fabricated habitat units designed specifically for ecological engineering have also been trialled. These aim to incorporate a number of different biodiversity enhancement features and may also perform a semi-structural function in developments. Notable examples include built-in (Perkol-Finkel and Sella, 2016) or retro-fitted (Browne and Chapman, 2014) rock pool units, BIOBLOCKS and similar breakwater units (Firth et al., 2014; Sella and Perkol-Finkel, 2015) and Reef Balls^{TM} (Harris, 2003; Reef Ball Foundation, 2016; Scyphers et al., 2015). Pre-fabricated ecological engineering units such as these may be the most effective and feasible means of building habitat complexity into marine developments at an ecologically-meaningful scale (i.e. to deliver tangible biodiversity enhancement). They could conceivably be mass-produced at a reasonable cost and incorporated into developments either during construction or retrospectively (see Seattle Seawall Project; Goff, 2010).

In the design of these units, material choice is an important factor. Concrete has been widely favoured because of its ease of casting into heterogeneous three-dimensional shapes and surface topographies. The net ecological benefits of enhancement using concrete, however, may be compromised for a number of reasons. Firstly, concrete has an enormous carbon footprint. Cement production alone has been estimated to account for around 6-7% of global anthropogenic CO_2 emissions (Meyer, 2009). Secondly, concrete production often requires an aggregate component, which again carries an environmental footprint (Flower and Sanjayan, 2007; Marinković et al., 2010), especially when sourced from the marine environment (Newell et al., 1998). Thirdly, high surface alkalinity (pH 12-13) and leaching of metals (McManus et al., this issue; Müllauer et al., 2015) can impair settlement of marine organisms, resulting in communities dominated by a few alkotolerant taxa such as barnacles (Dooley et al., 1999; Guilbeau et al., 2003). As such, communities that establish on concrete marine structures tend to differ from those found in natural habitats (Andersson et al., 2009; Glasby et al., 2007; Glasby and Connell, 1999; but see also Connell, 2000; Knott et al., 2004). Yet if these issues can be addressed, concrete holds huge potential for use in ecological engineering products.

So-called 'green' or 'eco' concretes have been developed and utilised in construction and civil engineering projects previously (Meyer, 2009; Perkol-Finkel and Sella, 2014, 2016; Sella and Perkol-Finkel, 2015). There is an extensive body of literature illustrating how the environmental footprint of concrete can be reduced through partial replacement of Portland cement (the primary source of CO_2 emissions in concrete production) and aggregates with:

pozzolanic industry by-products such as fly-ash, silica fume and ground granulated blast-furnace slag (GGBS) (Malhotra and Mehta, 1996; Meyer, 2009); waste materials such as shells, ceramic and end-of-life concrete (Cuadrado et al., 2015; Huang et al., 2004; Kuo et al., 2013; Marinković et al., 2010; Sekar et al., 2011; Yang et al., 2010, 2005); and natural fibres such as hemp and vegetable fibres (Awwad et al., 2012; Kidalova et al., 2012; Li et al., 2006; Pacheco-Torgal and Jalali, 2010; Pandey et al., 2010; Sedan et al., 2008). Pozzolanic industry by-products and other waste materials are often available at zero cost, but their rate of production exceeds their re-use. Hence, they are often disposed of in landfill or by incineration, at an economic and environmental cost (Cheerarot and Jaturapitakkul, 2004; Fry, 2012; Sekar et al., 2011). Pozzolans are capable of producing more chemically-resistant end-product concretes with reduced permeability and greater compressive strengths (Malhotra and Mehta, 1996; Meyer, 2009; Oner and Akyuz, 2007). They are, therefore, regarded as particularly suitable for applications in marine environments (Seleem et al., 2010). Natural fibres are a cheap and renewable resource with the capacity to sequester, rather than emit, carbon (Meyer, 2009). Furthermore, reinforcement with natural fibres – in particular hemp fibres – has been shown to increase the flexural strength of concrete materials (Awwad et al., 2012; Li et al., 2006; Merta and Tschegg, 2013; Sedan et al., 2008).

In some cases, such ‘green’ concretes have been employed in marine ecological engineering projects, with the aim of enhancing material properties for biodiversity (i.e. beyond considerations of the environmental footprint of production). The addition of pozzolans can reduce the surface pH of concretes (Fernández Bertos et al., 2004; Guilbeau et al., 2003; Park and Tia, 2004), potentially creating more favourable surfaces for colonisation by marine life (e.g. Nandakumar et al., 2003). Thus, Reef Balls^{TM} cast from concrete with microsilica additives (Reef Ball Foundation, 2016) have been deployed in artificial reef projects (Harris, 2003; Scyphers et al., 2015). Similarly, Econcrete^{TM} admixtures with different mixes of pozzolans have been used to make “ecologically active” concrete for both intertidal and subtidal coastal infrastructure developments (Perkol-Finkel and Sella, 2016, 2014; Sella and Perkol-Finkel, 2015). Waste mollusc shell material has also been incorporated into concrete marine structures to create textured surfaces and encourage gregarious settlement (Collins et al., 2015; Cuadrado et al., 2015; Ortego, 2006). Natural fibres, however, whilst suitable for incorporation in both structural (i.e. fibre-reinforced concrete: Awwad et al., 2012; Li et al., 2006; Sedan et al., 2008) and non-structural (i.e. ‘hempcrete’: Elfordy et al., 2008; Stanwix and Sparrow, 2014) building materials terrestrially, are not generally considered suitable for use in concrete in aquatic environments. This is due to durability concerns relating to increased permeability, reduced chemical attack resistance, dimensional instability and degradation of natural fibres (e.g. see Pacheco-Torgal and Jalali, 2010; Sivaraja et al. 2010). Yet there is evidence to suggest that the inclusion of pozzolans such as GGBS in the concrete matrix can counter these durability issues and prevent or delay internal fibre degradation (Pacheco-Torgal and Jalali, 2010; Pandey et al., 2010; Seleem et al., 2010). Natural fibre reinforced concretes may, after all, hold unrecognised potential for marine ecological engineering applications.

Evans et al. (2017) found that reducing the carbon footprint of engineered structures was prioritised (ranked in the top ten concerns) by stakeholders as a means of building-in secondary environmental benefits into marine developments. Although, as described above, a number of marine eco-engineering studies have trialled alternative ‘green’ concretes previously, there is limited published information available regarding the exact composition of these alternative materials, possibly on account of commercial sensitivities and IP protection. There is, therefore, great interest in investigating and reporting on additional alternative materials for marine ecological engineering applications.

We carried out a pilot study to trial alternative cast-able "*Reefcrete*" concrete mixes, with reduced environmental footprints, for use in marine eco-engineering products. Modifications were made with the objectives of developing materials that: (i) had a reduced carbon footprint compared to ordinary Portland cement based concrete; (ii) utilised natural and recycled materials as partial aggregate replacement; and (iii) provided suitable substrate for marine biodiversity to colonise. Given the known benefits of using pozzolanic industry by-products as cement replacements in concrete (Meyer, 2009), particularly in marine engineering (Nandakumar et al., 2003; Perkol-Finkel and Sella, 2016; Seleem et al., 2010; Sella and Perkol-Finkel, 2015), we used a GGBS based cement binder, with partial replacement of Portland cement with recycled GGBS, to achieve a reduced carbon footprint in our concrete mixes. To investigate the potential of achieving *further* reductions in environmental footprint, we trialled alternative GGBS based concrete mixes with partial replacements of coarse aggregate with hemp fibres and recycled shell material in varying proportions. We cast tiles from six alternative concrete blends – with different proportions of aggregate replacement by either hemp or shell – and compared them to standard GGBS concrete controls. We calculated the estimated carbon footprint of each concrete blend and deployed replicate tiles in the intertidal environment for 12 months to assess their performance as substrate for marine biodiversity. We tested null hypotheses that there would be no difference in the live cover, taxon richness or multivariate community compositions on each of the different types of concretes, including the control. Even a null result – indicating that these alternative materials provided substrate of equal habitat suitability to ordinary GGBS concrete – would support further development of their application on account of their reduced environmental footprints alone. We also compared initial algal recruitment and biofilm growth between concrete blends, to investigate whether any observed differences in assemblages recorded at the end of the study were evident during initial colonisation. We further assessed the beta-diversity of the total species pools utilising each different substrate, by comparing taxa accumulation over additional replicate tiles.

Given that concrete is the most commonly used material on earth (Gartner, 2004), this research will contribute to the rapidly expanding knowledge base on the ecological value of alternative materials for use in environmentally-sensitive construction practices.

2. Materials and methods

2.1 "Reefcrete" concrete blends

Six alternative "*Reefcrete*" concrete blends were trialled in this study and compared with a "GGBS Control" blend. The control blend comprised cement binder, fine aggregate (sharp sand sourced from Travis Perkins PLC.) and coarse aggregate (10 mm gravel/shingle sourced from Travis Perkins PLC.) in the ratio (by weight) 1 : 1½ : 2½, with a water : cement binder ratio of 0.4 (i.e. 1 : 2½). The cement binder consisted of a 70 : 30 mixture of ground granulated blast-furnace slag (GGBS) and Portland cement (CEM I) (both sourced from Ecocem Ireland Ltd.; http://www.ecocem.ie/).

Each of the six alternative concrete blends was the same as the "GGBS Control", but with varying percentage replacements of the coarse aggregate with either hemp fibres or crushed whelk shells (i.e. 'High', 'Medium' and 'Low' percentage replacements: see Table 1 for quantities). Hemp fibres and whelk shells were used as additional means of reducing the environmental footprint of the concrete blends through reduced requirement of extracted gravel aggregate and potential net carbon storage (see Section 2.2). Both alternative aggregates were also anticipated to alter the surface texture of the concrete when cast into tiles, compared to the control blend, although this was not controlled or measured (see Figure

1). Up to 100% coarse aggregate replacement by shell material was possible since it was similar in size, shape and volume to the conventional aggregate. Percentage replacement by hemp fibres, however, was restricted to a maximum of 25% of coarse aggregate weight, since beyond this the concrete lacked workability and strength.

Hemp fibres were sourced from KJ Voase & Son (http://www.eastyorkshirehemp.co.uk/). Fibres were separated from the woody stems (shives) of dried hemp plants and cut to ~10 mm lengths. To enhance fibre-cement bonding, the fibres were pre-treated according to Sedan et al. (2008), by soaking them for 48 hours in a 6% sodium hydroxide solution and rinsing with distilled water before drying at 60°C for 48 hours. Crushed waste whelk shells were sourced from a local seafood processing factory in West Wales (Quay Fresh & Frozen Foods Ltd.), where their licensed disposal onto the seabed is anecdotally thought to have detrimental impacts on beach amenity value and benthic habitats. Shells were washed and crushed to a size of approximately 10-20 mm, according to their disposal licence conditions.

Table 1 Cement binder ratios, aggregate replacement levels and carbon footprint estimates for one control and six alternative concrete blends. Carbon footprint estimates calculated by summing the estimated CO_2 emissions of their component parts, multiplied by their respective ratios within the blends (SOM Tables 1-3). Estimate is also given for ordinary Portland cement based concrete (CEM I Concrete) for comparison. Negative values indicate potential net carbon storage.

Blend	Cement binder ratio (GGBS : CEM I)	Alternative aggregate	Percentage aggregate replacement	Replicates (n)	Carbon footprint (kg CO_2/ t)
CEM I Concrete	0:100	n/a	n/a	n/a	189.84
GGBS Control	70:30	None	None	5	65.52
Low Shell	70:30	Shell	25%	3	53.44
Medium Shell	70:30	Shell	50%	3	41.35
High Shell	70:30	Shell	100%	3	17.18
Low Hemp	70:30	Hemp	5%	3	25.41
Medium Hemp	70:30	Hemp	10%	3	-14.70
High Hemp	70:30	Hemp	25%	3	-135.02

2.2 Carbon footprint calculations

Estimated carbon footprints (net embodied CO_2 emissions in kg per tonne of concrete) of each of the concrete blends trialled in this study were calculated (Table 1), by summing the estimated CO_2 emissions of their component parts, multiplied by their respective ratios within the blends (SOM Tables 1-3).

The carbon footprint of ordinary Portland cement (CEM I) based concrete was estimated, for comparison, to be approximately 190 kg CO_2 per tonne of concrete (Table 1; SOM Table 1). The estimated footprint of the "GGBS Control" blend used in this study was much lower on account of 70% replacement of CEM I with GGBS, which has an embodied carbon footprint up to 22 times lower than that of CEM I (Ecocem, 2016; Table 1; SOM Table 1). The footprints of the six alternative concrete blends were further reduced on account of reduced aggregate content and potential carbon storage in the hemp and shell material (Table 1; SOM Tables 1-3). The carbon content in hemp fibre accounts for 43.6% of its molecular mass, meaning that for each tonne produced, ~436 kg of carbon has been sequestered from the atmosphere (SOM Table 2). This carbon content was then converted to the equivalent in CO_2 by multiplying by the ratio of their respective molecular weights (IPCC, 2006). We thus estimated one tonne of hemp fibre to equate to 1.559 tonnes of absorbed CO_2 (SOM Table 2). Although the shell material used in this study was recycled from a process that would otherwise have resulted in licenced disposal onto the seabed, waste shells from the UK seafood processing industry are typically disposed of by landfill (78%) and incineration (22%) routes (Fry, 2012). Shell material sent to landfill would naturally persist for an extended period before decomposition and release of CO_2, therefore only the proportion that would otherwise be incinerated (i.e. with immediate CO_2 release), was used to calculate its potential carbon storage. The potential carbon storage was calculated by accounting for 95% $CaCO_3$ composition (White et al., 2007), with one gram of $CaCO_3$ containing 0.119 g carbon (by division of molecular weights). We thus estimated one tonne of shell material to equate to 0.091 tonnes of avoided CO_2 emissions (SOM Table 3). Additional CO_2 would also be sequestered by the carbonation process of concrete during curing (Galan et al., 2010); this was not included in calculations but was assumed to be constant across treatments.

2.3 Tile casting and deployment

The concrete blends were hand-mixed and cast into experimental tiles of 150 x 150 x 30 mm, with three replicate tiles per treatment (n = 5 for the "GGBS Control" since these were being utilised in a different study simultaneously). After curing for 14 days, the tiles were deployed in the intertidal zone at Aberystwyth, Wales, UK in October 2014 (Figure 1). The site is a moderately exposed macrotidal bedrock shore, surrounded by predominantly sedimentary habitats, interspersed by rocky outcrops and artificially hardened shorelines. The tiles were affixed to horizontal bedrock surfaces on the lower-mid shore and labelled with engraved washers. A radius of 50 cm around each tile was cleared of canopy macroalgae (mostly fucoids) in order to reduce disturbance from sweeping (Hawkins, 1983; Jenkins et al., 1999).

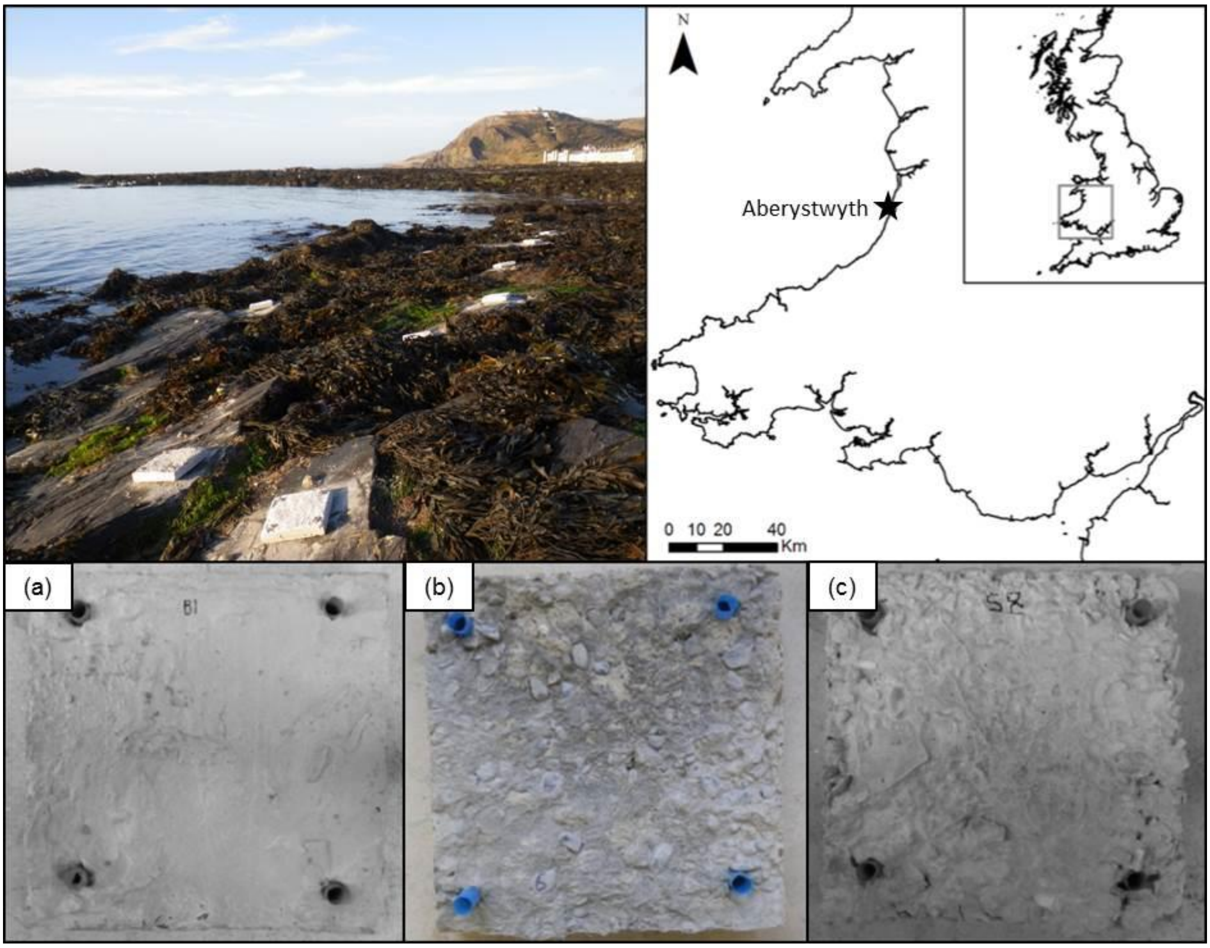

Figure 1 Experimental tiles deployed in the intertidal zone at Aberystwyth, Wales, UK (52°24'51.6"N, 4°05'27.9"W) alongside photographs illustrating apparent differences in surface texture on the top surfaces of tiles cast from (a) GGBS Control concrete blend, (b) hemp concrete blend, and (c) shell concrete blend, prior to deployment (Images: A. Evans, H. Dennis).

2.4. Data collection

A preliminary non-destructive survey of the tiles was undertaken *in situ* in December 2014 to record initial algal recruitment and biofilm growth. A bbe BenthoTorch^{TM} (Moldaenke, Germany) fluorometer was used to measure total chlorophyll-a concentrations along with individual concentrations of green algae, blue-green algae (cyanobacteria) and diatoms on the upper tile surfaces ($\pm$ 0.2 µg chl-a/cm^2). The means of four replicate measures were recorded for each tile to account for the inherent spatial variability of biofilm growth (Hutchinson et al., 2006; Jesus et al., 2005).

After 12 months, the tiles were recovered from the shore in October 2015 for laboratory analysis. One of the “Medium Shell” blend tiles was lost from the experiment, having been dislodged from the shore during stormy weather. Total percentage live cover of the upper tile surfaces was recorded from photographs (i.e. not taking account of understorey cover; maximum cover = 100%). All macroalgae and mobile fauna were then scraped from the upper tile surfaces for identification and enumeration. Macroalgae from each tile were identified, grouped by taxon and dried at 60°C for 48 hours. The biomass of each taxon was recorded for each tile by measuring dry weight (0.0001g). Mobile fauna from each tile were fixed in 70% ethanol and identified. Abundance counts of each taxon were recorded for each tile. Percentage cover of encrusting algae and fauna on the upper tile surfaces was also visually estimated. All organisms were recorded to species level, but where this was not possible, genus, family or consistent morphotaxa were used.

2.5 Statistical analyses

Initial algal concentrations (total algae, green algae, blue-green algae and diatoms; µg chl-a/cm^2), mean live cover (%), mean taxon richness (full community, sessile community and mobile community), taxa accumulation and multivariate community compositions (full community, sessile community and mobile community) were compared across the different types of concrete blends to assess their performance as substrates for marine biodiversity. The low level of replication employed (n = 3), however, prohibited rigorous statistical analyses of differences between each of the seven different blends trialled, especially since one of the “Medium Shell” tiles was lost from the experiment (n = 2). To increase statistical power, a subset of five tiles was selected from the “High” and “Medium” aggregate replacement blends of both the hemp and shell concretes (three “High Hemp” and two “Medium Hemp”; three “High Shell” and two “Medium Shell”), and were pooled for comparison with the five “GGBS Control” replicates. “High” and “Medium” replacement replicates were chosen to maximise the compositional difference (and thus associated environmental benefits) between alternative and control blends. Pooling over “High” and “Medium” replicates was possible since there was no significant difference in mean initial algal concentrations, mean live cover, mean richness or multivariate community compositions between “High”, “Medium” and “Low” percentage aggregate replacement blends for either hemp concretes or shell concretes (Kruskal Wallis and PERMANOVA tests: $p > 0.05$ in all cases; SOM Tables 4 and 5).

ANOVA was used to test for differences in mean initial algal concentrations (total algae, green algae, blue-green algae and diatoms), mean live cover and mean richness (full community, sessile community and mobile community separately) between the different concrete blends. Pairwise significant differences were identified using Tukey’s *post hoc* tests. PERMANOVA (based on Bray-Curtis similarity matrices and 9999 unrestricted permutations of raw data; Anderson 2001) was used to test for differences in multivariate community compositions (full community, sessile community and mobile community separately) between concrete blends. Pairwise tests were again employed to identify pairwise significant differences. For these *post hoc* tests, there were not enough possible permutations to perform a reasonable test of significance. Therefore, Monte Carlo *P* values were used as a more meaningful, but approximate, alternative (Anderson and Robinson, 2003). A one-way design was used for all analyses, with fixed factor Blend (three levels: “Hemp”, “Shell”, “GGBS Control”), and $n = 5$. SIMPER analysis (Clarke, 1993) was used to calculate percentage contributions of individual taxa to dissimilarities between communities recorded on the different concrete blends. Taxa accumulation curves (based on 9999 permutations of data) were plotted to investigate beta-diversity and the total species pools utilising the different concrete substrates.

Univariate analyses were carried out in SPSS version 23 (IBM Corp., Armonk, NY, USA) and multivariate analyses were carried out in PRIMER version 6 with PERMANOVA add-on (PRIMER-E Ltd, Plymouth, UK). To account for scale differences in abundance measures, data were fourth root transformed prior to multivariate analyses. Examination of Q-Q plots indicated that the data were approximately normally distributed. Levene's test and the PERMDISP routine (Anderson, 2006) confirmed homogeneity of variances and multivariate dispersions, respectively.

3. Results

Initial algal concentrations recorded on different concrete blends after 2 months were variable (Figure 2). Although mean chlorophyll-a concentrations for total algae, blue-green algae and diatoms were higher on the hemp concrete blends than on the shell and "GGBS Control" blends (Figure 2), differences were not significant (total algae $F_{2,12} = 2.80$; green algae $F_{2,12} = 0.36$; blue-green algae $F_{2,12} = 1.75$; diatoms $F_{2,12} = 3.14$; $p > 0.05$ in each case) (Figure 2).

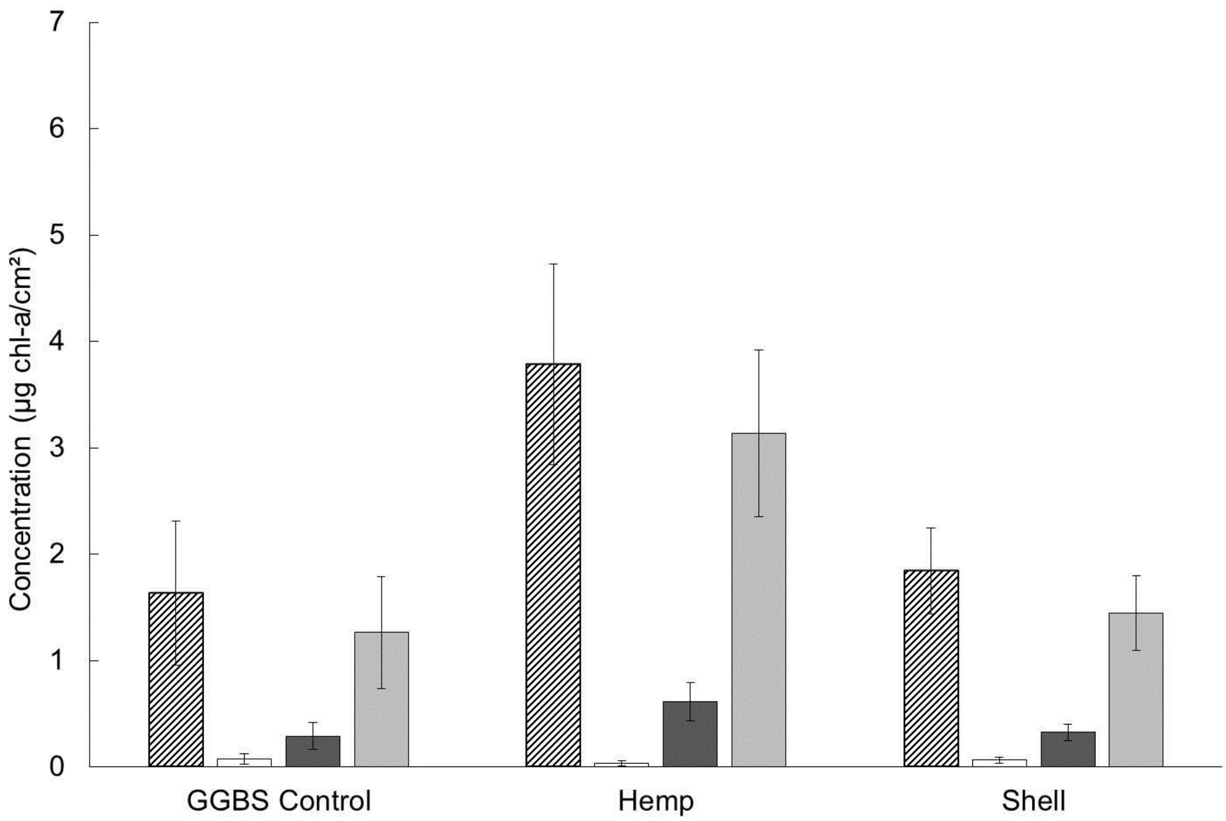

Figure 2 Mean (± SE) concentrations of total algae (hashed bars), green algae (white bars), blue-green algae (dark grey bars) and diatoms (pale grey bars) recorded after two months on tiles cast from alternative concrete blends with aggregate replacement by hemp fibres and shell, compared with a "GGBS Control" blend, n = 5.

After 12 months, mean live cover was significantly different between concrete blends ($F_{2,12}$ = 23.47, $p < 0.001$). *Post hoc* tests revealed that mean live cover was significantly higher on both the hemp (92.0% ± 3.7 SE) and shell (74.0% ± 15.9 SE) blends than on the "GGBS Control" blend (25.0% ± 13.6 SE) ($p < 0.001$), but there was no difference between the hemp and shell concretes ($p > 0.05$) (Figure 3). Mean taxon richness (full community) was also different between concrete blends, but significance was marginal ($F_{2,12}$ = 3.82, $p = 0.05$). Mean richness was higher on hemp blends (14.6 ± 1.9 SE) compared to both the shell blends (9.6 ± 1.6 SE) and the "GGBS Control" (8.8 ± 1.2 SE), but *post hoc* tests failed to report significant pairwise differences ($p > 0.05$ in each case, but $p = 0.06$ for hemp versus "GGBS Control"; Figure 3). When comparing mobile and sessile components of the communities separately, there was no significant difference in mean sessile taxon richness between the different concrete blends ($F_{2,12} = 1.55$, $p > 0.05$; Figure 3). The overall difference in richness was attributed, instead, to the mobile fauna components of communities ($F_{2,12} = 4.87$, $p < 0.05$) which were significantly richer on hemp blends compared to shell and “GGBS Control” blends ($p < 0.05$; Figure 3).

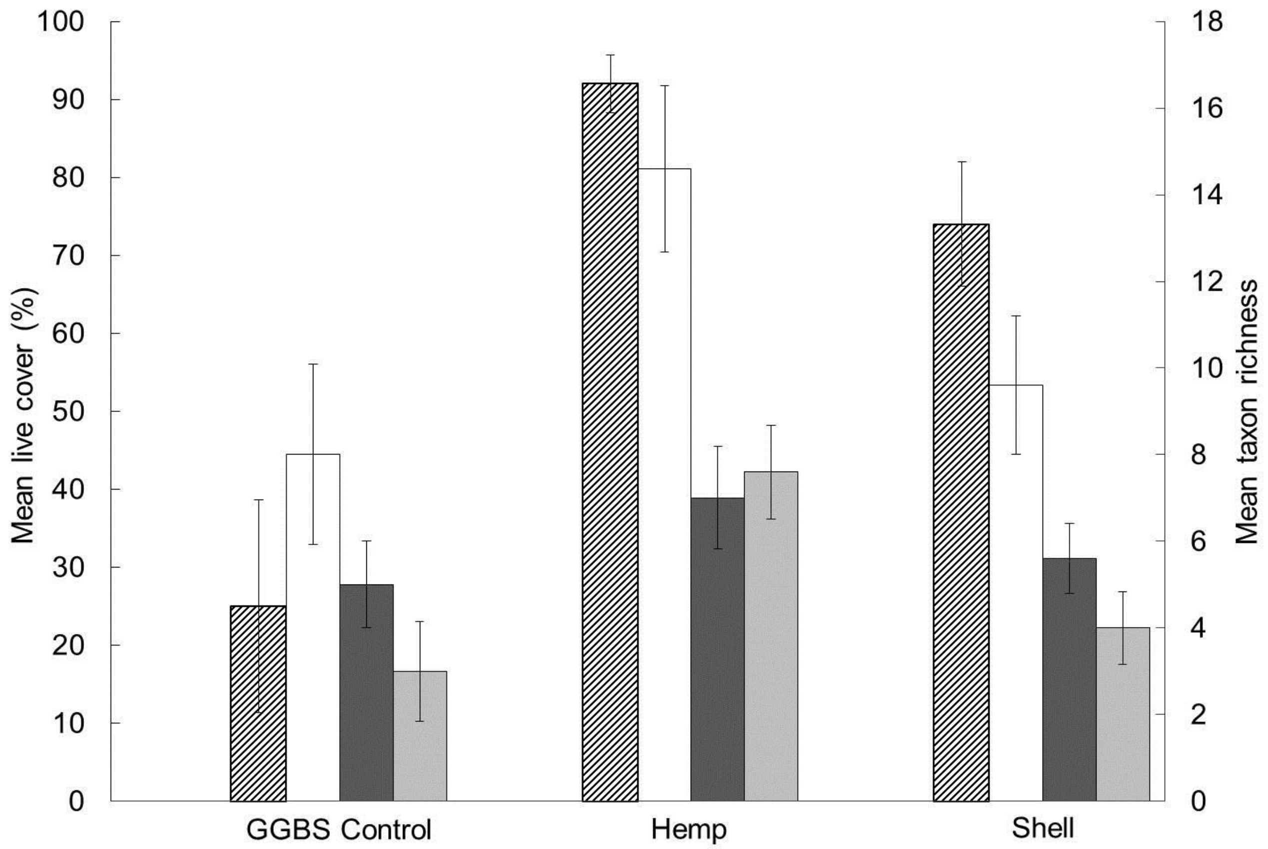

Figure 3 Mean (± SE) percentage live cover (hashed bars) and mean (± SE) taxon richness for full community (white bars), sessile community (dark grey bars) and mobile community (pale grey bars) recorded after 12 months on tiles cast from alternative concrete blends with aggregate replacement by hemp fibres and shell, compared with a “GGBS Control” blend, n = 5.

Multivariate community compositions were significantly different between concrete blends when comparing the full communities (Pseudo-$F_{2,12} = 2.19$, p(perm) < 0.05), and also when comparing the sessile (Pseudo-$F_{2,12} = 2.04$, p(perm) < 0.05) and mobile (Pseudo-$F_{2,12} = 2.15$, *p*(perm) < 0.05) components of the communities separately. *Post hoc* tests revealed that community compositions were in fact only different between the hemp and "GGBS Control" blends (*p*(mc) < 0.05), whereas no differences were found between the hemp and shell blends, or the shell and "GGBS Control" blends (*p*(mc) > 0.05) (Figure 4). SIMPER analysis reported that the differences between communities recorded on hemp concrete tiles and "GGBS Control" tiles were largely explained by higher abundances of most taxa (24 out of 30 taxa) on the hemp tiles – each contributing relatively little to overall dissimilarities – rather than by one or two dominant taxa (Table 2). Taxa that were more abundant on the hemp concrete tiles included the canopy alga, *Fucus serratus*, and several mobile grazers such as isopods, amphipods and *Patella vulgata* limpets (Table 2). Brown encrusting algae was notably more abundant on the "GGBS Control" tiles than on the hemp concrete tiles (Table 2). Twelve taxa that were recorded on the hemp concrete tiles were absent from the "GGBS Control" tiles, including blue mussels, *Mytilus edulis*, the reef-building worm, *Sabellaria alveolata*, and a number of isopods and littorinid snails (Table 2). Hemp concrete tiles collectively, therefore, supported increased diversity and abundance of taxa compared to "GGBS Controls".

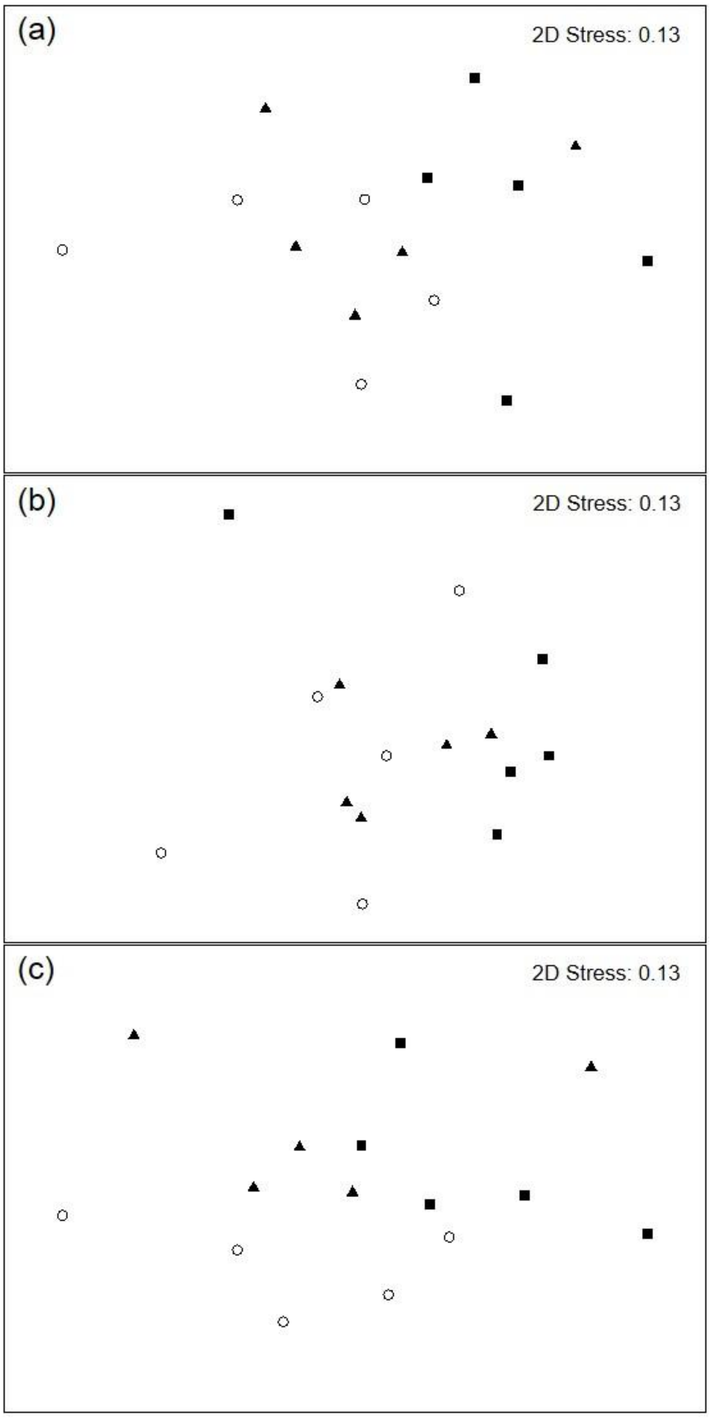

Figure 4 nMDS ordinations of (a) full communities, (b) sessile communities, and (c) mobile communities recorded after 12 months on tiles cast from alternative concrete blends containing aggregate replacement by hemp fibres (squares) and shell (triangles), compared with a “GGBS Control” blend (circles), n = 5.

Table 2 Differences (< and >) in mean abundances (biomass (g), counts (c) or percentage cover (%) of taxa recorded after 12 months on tiles cast from alternative concrete blends containing aggregate replacement by hemp fibres and a "GGBS Control" blend, n = 5.

%: percent contribution to multivariate dissimilarity; Diss/SD: dissimilarity divided by standard deviation of contributions across all pairs of samples (measure of consistency of contribution)

Average dissimilarity = 55.71%

Species	**GGBS Control**		**Hemp**	**%**	**Diss/SD**
Fucus serratus (g)	0.58	<	17.94	8.64	1.78
Idotea granulosa (c)	1.20	<	7.00	6.58	1.19
Mytilus edulis (c)	0	<	5.40	6.41	1.16
Brown encrusting algae (%)	23.60	>	4.80	6.06	0.88
Gammaridae sp. 1 (c)	6.20	>	1.40	5.24	1.33
Patella vulgata (c)	0.60	<	2.00	5.22	1.26
Amphipoda (c)	1.80	<	3.40	4.83	1.13
Ulva spp. (g)	2.32	<	2.62	4.43	1.25
Melarhaphe neretoides (c)	0	<	1.20	4.43	1.15
Spirorbis spp. (%)	1.60	<	2.50	4.34	1.18
Sphaeromatidae (c)	0	<	0.80	4.31	1.14
Fucus spp. (juv.) (g)	0.21	<	1.16	3.62	1.84
Gibbula umbilicalis (c)	1.40	<	2.00	3.40	0.91
Spirobranchus spp. (%)	0.60	<	0.80	3.29	0.88
Porcellana platycheles (c)	0.60	>	0.20	2.97	0.85
Sabellaria alveolata (%)	0	<	1.40	2.83	0.79
Littorina obtusata (c)	0	<	0.40	2.82	0.75
Bryozoan crust (%)	0.60	<	1.00	2.67	0.68
Palmaria palmata (g)	0.06	<	0.08	2.41	1.59
Carcinus maenas (c)	0	<	0.60	2.31	0.79
Littorina saxatilis (c)	0	<	0.40	2.11	0.79
Dynamene bidentata (c)	0.40	>	0.20	2.06	0.67
Chthamalus spp. (c)	0.20	>	0	1.65	0.47
Porphyra spp. (g)	0.02	<	0.06	1.51	0.67
Perinereis cultrifera (c)	0.20	>	0	1.22	0.48
Asterina gibbosa (c)	0	<	0.20	1.06	0.49
Janiridae (c)	0	<	0.20	1.05	0.49
Isopoda (c)	0	<	0.20	10.5	0.49
Crustacea (c)	0	<	0.20	1.05	0.49
Polysiphonia spp. (g)	0	<	0.01	0.41	0.49

Taxa accumulation curves indicated that the total species pool recorded on the hemp concrete blend tiles (28 taxa) was much larger than that recorded on the shell concrete tiles and the "GGBS Control" tiles (18 taxa in both cases; Figure 5).

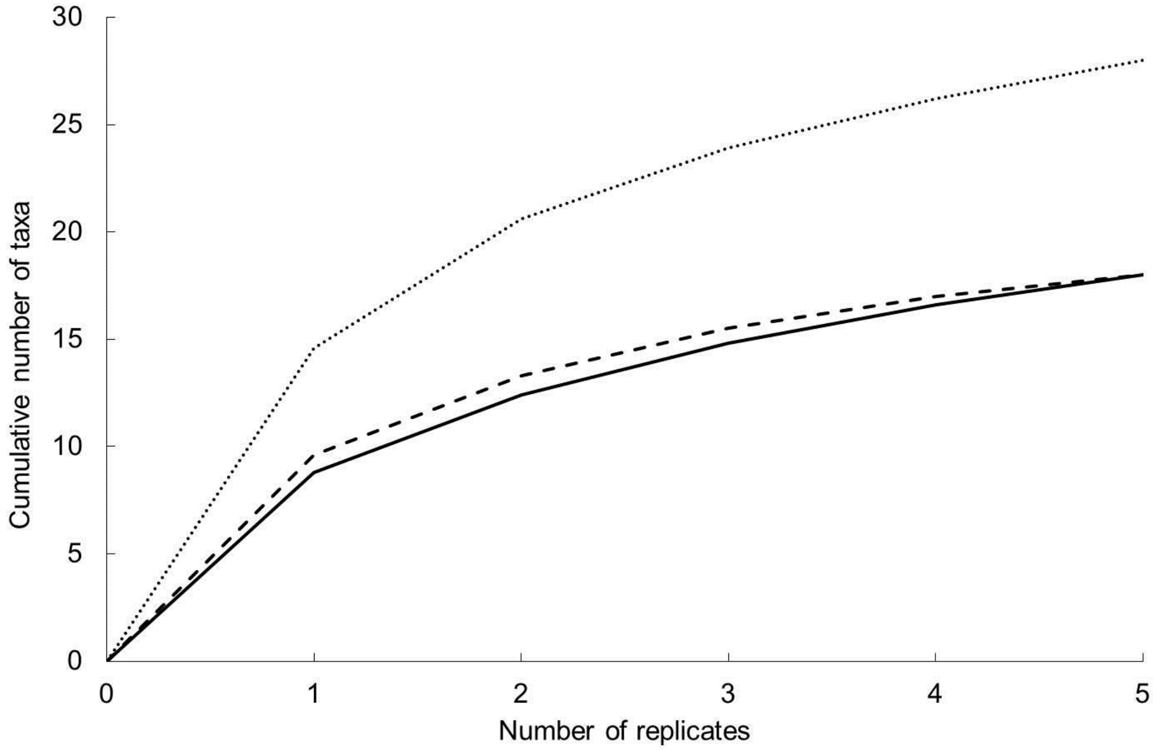

Figure 5 Cumulative number of taxa recorded after 12 months on five replicate tiles cast from alternative concrete blends containing aggregate replacement by hemp fibres (dotted line) and shell (dashed line), compared with a "GGBS Control" blend (solid line).

4. Discussion

Here we have demonstrated the potential to reduce the environmental footprint of concrete, beyond replacement of Portland cement with recycled GGBS, by partially replacing the coarse aggregate component with hemp fibres and recycled shell material. Although this in itself is not new – similar alternative 'green' concretes have been extensively studied previously (Meyer, 2009) – in this study we provide novel evidence that such hemp and shell concrete blends can provide substrate of equal or better habitat suitability for intertidal biodiversity compared to standard GGBS based concrete. Furthermore, although we did not test the engineering properties (e.g. strength, elasticity, hardness, workability) of the blends in this pilot study, the tiles persisted for 12 months in the intertidal environment despite a series of storms during winter 2014. Only one tile was lost from the experiment. This may have been a result of a weakness in the concrete matrix, but alternatively may have been due to a weak fitting or erosion of the bedrock on which it was attached. We therefore propose that, although natural fibre reinforced concretes are not generally considered suitable for use in aquatic environments (see discussion of literature in the Introduction), the materials trialled in this study and their engineering properties are worthy of further investigation and consideration for use in ecological engineering products for marine developments.

4.1 Environmental footprint reductions

The environmental footprint of the GGBS based concrete control blend used in this study was already substantially lower than that of ordinary Portland cement based concrete (estimated carbon footprint in kg CO_2 per tonne of concrete reduced by 65.5%; see Section 2.2). By reducing the proportion of extracted coarse aggregate (Flower and Sanjayan, 2007; Marinković et al., 2010) and accounting for potential carbon storage in natural fibres and shell material (see references in Section 2.2), our experimental hemp and shell concrete blends achieved considerable further reductions. The carbon footprints were estimated to be reduced by 18-74% with aggregate replacement by shell material (72-91% reduction compared to Portland cement based concrete), and by 61-305% with aggregate replacement by hemp (87-171% reduction compared to Portland cement based concrete), which made this concrete blend carbon negative.

In the context of pre-cast habitat units, the amount of concrete required for (and thus the carbon footprint of) a particular eco-engineering intervention would depend on the overall management objectives and the scale of the development in which units were to be deployed. As an example, standard Reef Balls^{TM} comprise approximately 1.9 tonnes of concrete per unit (Reef Ball Foundation, 2007) and BIOBLOCKS comprise approximately 5.4 tonnes per unit (Firth et al., 2014). Multiple units would almost certainly be required to achieve tangible biodiversity benefits at a meaningful scale in any given development. To date, most interventions have been implemented at an experimental scale only (e.g. Browne and Chapman, 2014; Chapman and Blockley, 2009; Evans et al., 2016; Firth et al., 2014, 2016a; Loke and Todd, 2016). When implemented in practice – for example, to satisfy licensing conditions for a development – the number of units required to deliver tangible benefits is likely to be in the hundreds for larger km-scale developments. Cumulatively, this would result in a substantial environmental footprint, which would compromise the intended environmental benefit of the intervention. Utilising concrete with a reduced or negative carbon footprint may be essential to counter this concern.

4.2 Substrate for marine biodiversity

After 12 months deployment in the intertidal environment, the hemp and shell concrete tiles supported significantly higher live cover of marine organisms than the GGBS concrete controls. Furthermore, tiles with hemp fibres supported greater taxon richness, particularly of mobile fauna, than the controls, and a much larger species pool overall. They also supported higher abundances of most taxa, in particular the canopy-forming macroalga, *Fucus serratus*, and several mobile grazers.

Colonisation of hard substrata in the marine environment can be influenced by the geology (Coombes et al., 2011; Green et al., 2012) and chemistry (Railkin, 2004; Roberts et al., 1991) of the material, and the texture of the substratum surface (Coombes et al., 2015; Köhler et al., 1999; Paalvast, 2015). Variations in concrete surface chemistry, such as leaching rates of metals and alkalis, are known to affect settlement of certain species (Dooley et al., 1999; Guilbeau et al., 2003; Nandakumar et al. 2003), while increased nutrient levels at the substrate-water interface may promote microalgal growth (Hillebrand and Sommer, 1997; Meyer-Reil and Köster, 2000, but see Thompson et al., 2004). Increased surface rugosity has been reported to promote initial recruitment of biofilms (Hutchinson et al., 2006; Kerr et al., 1999; Köhler et al., 1999 but see Sweat and Johnson, 2013), macroalgae (Fletcher and Callow, 1992; Harlin and Lindbergh, 1977; Hutchinson et al., 2006; Johnson, 1994) and invertebrate larvae (Coombes et al., 2015; Köhler et al., 1999; Walters and Wethey, 1996, but see Berntsson et al., 2000), via both active selection (Berntsson et al., 2000; Köhler et al., 1999) and passive (Köhler et al., 1999) mechanisms. Relationships are not, however, always linear (Hutchinson et al., 2006; Köhler et al., 1999; Sweat and Johnson, 2013; Walters and Wethey, 1996). Rugosity can subsequently influence the outcomes of biological interactions such as competition (Walters and Wethey, 1986), grazing (Hutchinson et al., 2006; Lubchenco, 1983) and predation (Johnson et al., 1998; Walters, 1992) post-settlement. Although we did not measure or control differences in chemistry or surface rugosity in this pilot study, it was apparent that the aggregate replacement by hemp fibres and shell fragments created more textured surfaces on the tops of the experimental tiles than the control concrete (Figure 1). In addition, in the hemp concrete blends, some small fragments of hemp fibre were noted protruding from the concrete surface. These would have been broken down over time, potentially increasing nutrient availability and bacterial concentrations on the tile surfaces. It is likely that these were the factors that explained the differences in colonisation between the different concrete blends observed in this study, since all the tiles were deployed at a similar shore height at the same location (i.e. they were subject to the same environmental context).

Initial biofilm concentrations after two months did not differ significantly between the different concrete blends, but mean concentrations of total algae, diatoms and blue-green algae tended to be higher on the hemp tiles. This may, to some extent, have influenced subsequent recruitment and colonisation trajectories (Park et al., 2011; Qian et al., 2007; Thompson et al., 1998; Wieczorek and Todd, 1997, 1998). Biofilms are, however, inherently spatially and temporally variable (Hutchinson et al., 2006; Jesus et al., 2005) and it was not possible to infer any direct effect of biofilm recruitment on successional community development in this pilot study.

It would be reasonable to assume that the higher live cover recorded on the hemp and shell concrete blends compared to the control, was a result of the rougher surface textures and potentially different chemistry at the substrate-water interface. The higher diversity of mobile fauna on the hemp tiles, however, was probably also due to the higher abundances of *Fucus* canopy algae, providing a source of food and habitat (Chemello and Milazzo, 2002; Fredriksen et al., 2005; Thompson et al., 1996). The role of canopies in facilitating diverse intertidal communities is well recognised (Eriksson et al., 2006; Thompson et al., 1996; Watt and Scrosati, 2013). These habitat engineers are expected to become even more important with increased likelihood of heatwaves in the future (IPCC, 2014) because of their capacity to stabilise the microclimate and ameliorate habitat conditions beneath canopies during the tide-out phase (Coombes et al., 2013; Moore et al., 2007; Watt and Scrosati, 2013; discussed in more detail in Hawkins et al., 2016). This stabilising function may also “bioprotect” engineered structures from weathering and erosion, thus enhancing the durability of construction materials (Coombes et al., 2013).

Canopy algae are suffering widespread declines in many parts of the world (Airoldi and Beck, 2007; Connell et al., 2008; Krumhansl et al. 2016; Mangialajo et al., 2008; Mineur et al., 2015). As natural intertidal rocky habitats are increasingly "squeezed" by sea level rise (Jackson and McIlvenny, 2011), artificial structures may become important surrogate habitats for them and other rocky shore organisms (see Evans, 2016). Engineered marine structures, however, tend to support lower algal canopy cover compared to natural rocky reefs (Ferrario et al., 2016; Glasby, 1999; Moschella et al., 2005). In the Adriatic Sea, there have even been efforts to transplant fucoid canopies on to coastal defence breakwaters to aid their conservation (Perkol-Finkel et al., 2012). Although physical factors alone are unlikely to be the only reason for reduced success of canopy algae on artificial structures (Ferrario et al., 2016), there would certainly be value in identifying alternative construction materials that may promote higher canopy cover. Furthermore, macroalgae are highly productive and may have an important, as yet unrecognised, role in carbon storage and sequestration (Krause-Jensen and Duarte, 2016; Smale et al., 2016). If hemp concretes can support more macroalgae than ordinary concrete then this may be considered an additional carbon-saving in the evaluation of its footprint as a construction material.

Biodiversity and community structure were not directly compared between our "*Reefcrete*" tiles and the adjacent natural rocky shore. Qualitative comparison with data collected in previous surveys (P. Moore, unpublished data), however, suggests that the communities colonising the experimental tiles were representative of the broader species pool. In addition, no non-natives were recorded on any of the tiles.

4.3 Next steps

Although our findings demonstrate ecological potential in the alternative concretes trialled for use in blue-green engineering of marine structures, this was very much a pilot study with a number of limitations. First, the low replication employed did not provide sufficient power to compare each of the seven different concrete blends separately, to assess the effect of differing levels of aggregate replacement by hemp or shell (especially since one of the tiles was lost from the experiment). Clearly, by incorporating higher proportions of the alternative aggregates, it was possible to achieve greater carbon footprint savings (see Section 2.2), but it was not possible to determine whether there would be any beneficial or detrimental effect on the suitability of the material for marine biodiversity. Second, we did not formally test the engineering properties of the blends for compliance to specified requirements of structural marine concrete (e.g. BSI and ASTM standards for compressive strength, volumetric stability, permeability and sulphate attack resistance). This would be an essential next step in evaluating their potential for use in marine engineering. It may be necessary to adjust mix ratios and utilise more sophisticated blending and casting techniques to achieve industry standards. It may also prove useful to explore options regarding preparation of the alternative aggregates (e.g. treatment of fibres to improve degradation resistance: Wei and Meyer, 2014; and shell fragments crushed to different sizes: Cuadrado et al., 2015). It is important to note, however, that for certain *non-structural* ecological engineering purposes, such as casting retrofit habitat units, it may not be critical to meet the same standards as materials that are used to load-bear or maintain structural integrity. It is, however, important that materials in habitat units will persist in the environmental context for which they are intended. Third, the carbon footprint calculations presented here are preliminary estimates only, based on the logic and values cited in the literature (see Section 2.2). There may, however, be additional considerations that affect the overall environmental footprints of different concrete blends. In particular, it would be necessary to more closely investigate the efficiency of hemp fibre carbon storage, with quantification of remineralisation rates through erosion or the actions of microbial cellulose digestion (e.g. see Gu et al., 1998).

The next step for this research, then, will be to scale up trials (spatially and temporally) to test novel “*Reefcrete*” concrete blends cast into different habitat enhancement units to be deployed in both intertidal and subtidal engineered marine developments. Their potential to deliver tangible and lasting enhancement to biodiversity on structures could then be assessed. Quantitative comparisons with communities colonising adjacent natural rocky habitats would further reveal their potential to provide surrogate habitat for rocky shore and reef biodiversity.

5. Conclusion

Concrete habitat enhancement units hold great potential for eco-engineering of artificial marine structures (Browne and Chapman, 2014; Firth et al., 2014; Goff, 2010; Harris, 2003; Perkol-Finkel and Sella, 2016; Scyphers et al., 2015; Sella and Perkol-Finkel, 2015). Concrete is a particularly useful material for this purpose on account of its low cost and ease of casting into textured surfaces and heterogeneous three-dimensional shapes. It must be acknowledged, however, that the negative environmental impacts from production (e.g. CO_2 emissions, aggregate extraction; Meyer, 2009) and deployment (e.g. habitat loss, biotic homogenisation; Firth et al., 2016b) of additional non-essential concrete structures in the marine environment may outweigh the potential ecological benefits. ‘Green’ or ‘eco’ concretes, using recycled industry by-products such as GGBS as cement binder, have been trialled to reduce the environmental footprint of units and to make them more suitable for marine biodiversity to colonise (McManus et al., this issue; Nandakumar et al., 2003; Perkol-Finkel and Sella, 2014; Sella and Perkol-Finkel, 2015).

We have shown here that it is possible to *further* reduce the environmental footprint of concrete (making it carbon-negative in some cases) by also partially replacing the coarse aggregate component of the matrix with hemp fibres or waste shell material. The hemp and shell concretes that we trialled proved to be of equal or greater suitability for marine biodiversity to colonise compared to ordinary GGBS based concrete. We propose that these novel "*Reefcrete*" concretes may hold great eco-engineering potential to address some of the key issues related to burgeoning ocean sprawl that is impacting marine and coastal ecosystems globally.

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